Frequent coppicing deteriorates the conservation status of black alder forests in the Po plain (northern Italy)

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Highlights

- Alluvial forests with black alder are a priority conservation habitat in Europe
- We assess whether frequent coppicing is compatible with their conservation
- We measured the effect of time since coppicing on forest structure and diversity
- Coppicing simplified vertical stand structure and the herbaceous layer for 20-30 years
- Non-native species were significantly more abundant in recently coppiced areas.

[1] GV wrote the paper and carried out statistical analyses, FM and GC carried out field sampling and forest structure analyses, MFe designed and coordinated the research and carried out imagery analyses, MFr carried out soil sampling and analyses, RM provided input for study design, interpretation and discussions, and ML carried out phytosociological analyses.

Abstract

Alluvial forests with black alder are a priority conservation habitat in Europe. In the Po plain, black alder is traditionally managed by coppicing with frequent rotations. This study aims to ascertain whether such management is compatible with habitat conservation, by measuring the effect of time since coppicing on forest structure and plant species composition across different layers.

We compared the effects of three treatments, each thrice replicated: recent (10-20 years), medium (20-30 years) and old coppice (>40 years). In all nine stands we measured basal area, tree and regeneration density, mean tree diameter and height, dominance by alder, species richness, Shannon diversity, and the number of ruderal and non-native species. Significant differences in dendrometric variables, species richness, diversity, and percent cover by chorotype were assessed for treatment effects by two-way ANOVA.

Frequently coppiced stands had a lower basal area, mean tree size, and volume, a more simplified vertical structure, a lower cover of the herbaceous layer and higher bare soil cover due to harvesting disturbance, a significantly lower cover by typical woodland *Fraxinetalia* species, and a significantly higher frequency and cover of non-native species.

Our study showed that frequent coppicing worsened the conservation status of black alder forests in the study area, simplified stand structure, deteriorated species composition, and increased the spread of non-native and ruderal plant species. Such negative effects persisted even 20-30 years after cutting. We recommend amending the current legislation and introducing mandatory Implications Assessment procedures everywhere alder forests are susceptible to be impacted in a similarly negative way.

Keywords: coppice, floodplain forests, forest management, Habitat Directive, non-native species, plant diversity, understory

1. Introduction

Black alder (*Alnus glutinosa* (L.) Gaertn.) is a tree species of riparian and water-logged habitats that is naturally widespread from mid-Scandinavia to southern Europe (Kajba et al. 2003). It forms pure stands on periodically submerged sites, while it mixes with ash (*Fraxinus excelsior* L.), maples (*Acer pseudoplatanus* L. and *Acer platanoides* L.) and oaks (mostly *Quercus robur* L.) on riverside and plateau sites (Dethioux 1974), where its intolerance to shading and lower groundwater tables reduce its ability to compete (Claessens et al. 2010).

Black alder grows between sea level and 1,300 on the Alps (Shaw et al. 2014). It is largely indifferent to soil parent material, but it requires precipitation above 510 mm per year and high water saturation (McVean 1953), and a high degree of atmospheric humidity throughout its reproductive cycle. When the water-table sinks below the surface during summer, tree growth increases but seedlings may suffer from drought (McVean 1953). The tree is able to fix atmospheric nitrogen in symbiotic root nodules (Bond et al. 1954), and its litter increases nitrogen and phosphorous content of the soil (Moiroud 1991, Giardina et al. 1995). The species has a maximum lifespan of 100 to 160 years (Claessens et al. 2010). It reaches sexual maturity at age 3-30, when it starts producing seeds with mast pulses every 3-4 years (Dethioux 1974). Seeds are dispersed by water or wind (up to 150 m: McVean 1955, but usually within 30 m: Funk 1990). However, regeneration occurs mostly from vegetative reproduction, e.g. in linear flood populations (Koop 1987, Deiller et al. 2003). Regeneration from seed is usually scattered and it occurs under favorable establishment conditions, e.g., on low-lying alluvial land or on former meadows (Douda et al. 2009). Seedlings require a higher

light intensity than those of larger-seeded trees (McVean 1956); it was found that natural regeneration of black alder is not possible under the canopy of a mature stand (Tapper 1993), except in openings larger than 0.1 ha (Claessens et al. 2010). The regeneration of black alder also depends on the frequency and intensity of disturbance (e.g. browsers, floods, or forest harvesting) (Pokorný et al. 2000; Wolf et al. 2004), and on the abundance of herbs that may compete with the seedlings (McVean 1956).

Due to their specific hydrological regime and rare occurrence, black alder forests and carrs are considered an endangered forest community in Europe (Ellenberg 1996). Alluvial forests with black alder and ash are a priority habitat of Community interest listed in the Annex I of the Habitats Directive 92/43/EEC as 91E0* – Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (*Alno-Padion*, *Alnion incanae*, *Salicion albae*). These forests are highly important for the conservation of a great number of typical woodland and floodplain plant species (Claessens 2003), particularly when interspersed in an agricultural matrix. Despite being often small and fragmented (Schnitzler 1994), black alder forests are often characterized by a high richness in herbaceous species (Brown et al. 1997). Beyond plant diversity, black alder forests support other ecosystem services as well, such as water filtration and purification in waterlogged soils (Peterjohn and Correll 1984), flood control and riverbank stabilization (Piégay et al. 2003).

Currently, these forests represent less than 1% of the forest cover in most European countries (Claessens et al. 2010) due to both land use changes such as conversion to non-native tree plantations or agricultural land, or to environmental changes related to human activities, e.g. land draining, impact of industrial areas, negative selection in favor of more valuable timber species such as oak and ash, and the introduction of non-native species (EEA 2012). For

these reasons, the conservation status of 91E0* habitat is currently “unfavourable inadequate” or “unfavourable bad” (Kremer et al. 2015).

In the Po plain, black alder is traditionally managed by coppicing, with rotations of 10 to 30 years due to the fact that the potential for vegetative regeneration from stumps declines at 60-80 years of age (Kapustinskaite 1960). Private ownership usually prevails in floodplain forests, with the consequence of creating a mosaic of small but intense and frequent cuttings, which can deteriorate habitat conservation and spatial continuity. Moreover, floodplain forests are highly vulnerable to plant invasions due to the frequent and intense natural disturbances, to their linear nature which facilitates long-distance species dispersal, and to intensive human pressure (Richardson et al. 2007). In this perspective, the question arises whether such management is compatible with habitat conservation.

This study aims to ascertain the effect of time since coppicing on the conservation status of black alder stands, as measured by (a) forest composition, structure, and biomass, and (b) species composition and naturalness of the herbaceous layer, e.g., the relative frequency of ruderal and non-native herbaceous species.

2. Study area

The study was conducted in the Natura 2000 site "IT1110021 - Laghi di Ivrea" (Figure 1), a 1600-ha Site of Community Importance (SCI) at the center of the 500-km^2 Ivrea Morainic Amphitheater (IMA). Mean annual temperature and annual precipitation are 12.5°C and 1002 mm, respectively (years 1921-2000) (Andreone et al. 2001). The bedrock is a juxtapositions of three metamorphic units (eclogitic micaschists, basic granulites and vulcanites) as a result

of uplift and underplating during the Tertiary Alpine orogenesis (Johnson 1973). The Morainic Amphitheater was constructed between the Pleistocene and the Last Glacial Maximum (Carraro et al. 1974). Thereafter, small lakes formed in the gaps between secondary moraines, but most later evolved into peat bogs or were artificially drained. Such low-elevation sites are characterized today by Endoaquepts or Haplosaprists soils (Piazzi et al. 2007). The latter is predominant in peat and raised bogs, where the sapric organic material has an extremely slow hydraulic conductivity and C/N ratios may be as high as 45.

The site hosts 11 habitat types of the EU Habitats Directive (1992/43/EEC Annex I), among which the priority habitat 91E0* covers 59 hectares. A total of 32 plant and animal species of the EU Nature Directives (1992/43/EEC and 2009/147/EC Annex II) (Natura 2000 Network Viewer 2016). Anthropic pressure has caused the number of plant species to decline from 179 to 160 species between 1950 and 2005; at least 12 non-native plant species were reported in the area so far (Minuzzo et al. 2005; Lonati et al. 2014).

Forests are mostly owned by small private owners. Between January 2012 and June 2015, 40 silvicultural treatments were authorized across 8 hectares of 91E0* forests inside the site; 10% of this area was treated by thinning, 30% by coppicing, and 60% by contemporary cutting of the coppice and high forest layers (Regione Piemonte 2016a).

3. Methods

We designed the study as a chronosequence of stands coppiced in three different times: recent (10-20 years, TR1), medium (20-30 years, TR2) and old coppicing (>40 years, TR0). To do so, we preliminarily assigned one of such treatments to all forest stands classified as 91E0*

146 habitats (according to Andreone et al. 2001) within the study area, based on the analysis of
147 repeated aerial images (years 1954, 1975, 1979, 1994-1996, 1998-1999, 2007, 2009). The
148 images were orthorectified and georeferenced, then visually classified into forested /
149 nonforested categories, and differentiated to obtain age ranges for each forest stand. Age
150 classes were subsequently confirmed by field surveys and exploratory increment core
151 sampling. Only stands belonging to the association *Carici remotae-Fraxinetum* Koch ex
152 Faber 1926 (alliance *Alnion incanae* Pawłowski in Pawłowski and Wallisch 1928) and
153 already existing in year 1954 were considered for further analysis, i.e., waterlogged stands of
154 the alliance *Alnion glutinosae* Malcuit 1929 and secondary stands on former non-forested
155 land were filtered out.
156
157 Following superposition to cadastrial stand maps, we identified three independent study areas
158 where all three elements of the chronosequence could be found in stands less than 100 m
159 apart from one another, in order to minimize site differences between treatments and counter
160 pseudoreplication. The only three areas where this condition was met in all the SCI are
161 indicated in Figure 1. A total of nine stands (i.e., 3 study areas x 3 treatments) were selected
162 for analysis; stands were at a constant elevation (about 240 m a.s.l.) and had a mean size of
163 1120 m^2.
164
165 In spring 2015, in each stand we randomly established a circular sampling plot (radius = 10
166 m) where we recorded species, frequency, diameter at breast heigh (dbh), origin (seed or
167 sucker) and height of all adult trees with dbh >=7.5 cm. We also recorded species, frequency,
168 origin, and height of all juvenile trees (dbh <7.5 cm) in a concentric 6-m radius circular plot.
169 From plot data we computed common descriptors of stand structure (species composition,
170 number of trees per hectare, basal area, quadratic mean diameter, average and top height,

171 percent trees originated from seed) and compared them across treatments by Mann-Whitney
172 test.
173
174 Within each sampling plot we randomly established five understory subplots (radius = 2 m)
175 where we visually assessed percent cover of upper tree (height >15 m), lower tree (height
176 between 5 and 15 m), upper shrub (height between 1.3 and 5 m), lower shrub, herbaceous and
177 bare soil layers, and assigned cover-abundance scores (Braun-Blanquet 1932) to all plant
178 species by each layer. At the center of each understory subplot, we measured canopy cover
179 by taking a hemispherical photograph at 1m height above the ground. Hemispherical
180 photographs were shot in Nikon .NEF format at 400 ISO with a 6 Megapixel Nikon D70S
181 equipped with a Samyang 8mm f/3.5 aspherical IF MC Fisheye Lens set at shutter priority
182 (time = 1/500 s). Canopy cover was determined by averaging the ratio of white to total image
183 pixels obtained by each of four global thresholding algorithms (Otsu 1979; Huang and Wang
184 1995; Yen et al. 1995; Li and Tam 1998) for the Fiji image analysis software (Schindelin et
185 al. 2012), applied to the blue band of each photograph.
186
187 In order to check for the absence of significant edaphic differences, three topsoil samples
188 were extracted from the center of each understory subplot at a depth of 0-10 cm. All samples
189 were air-dried and sieved (< 2 mm). Total carbon (corresponding to total organic carbon,
190 TOC, thanks to the absence of carbonates) and nitrogen (TN) were analyzed by dry
191 combustion with a CN elemental analyzer (CE Instruments NA2100, Rodano, Italy).
192
193 For each understory subplot we computed species richness (total number of species), the
194 Shannon diversity index, and the number of non-native species based on chorotype
195 (according to Celesti-Grapow et al. 2009). We associated a phytosociological optimum

196 (according to Aeschimann et al. 2004) to each vascular plant species at the class level,
197 including all subordinated syntaxa (Lonati et al. 2013; Orlandi et al. 2016; Pittarello et al.
198 2016) (Online Resource 1), and computed the number of species belonging to the following
199 phytosociological groups, listed in order of increasing conservation value: ruderals (classes
200 *Artemisietea vulgaris* and *Bidentetetea tripartitae*), tall herbs (classes *Filpendulo-*
201 *Convolvuletea* and *Molinio-Arrhenatheretea*), shrubs (classes *Crataego-Prunetea*,
202 *Franguletea* and *Salicetea purpureae*), and European alder forests (order *Fraxinetalia*). After
203 converting cover-abundance data to mid-percent values ('+' in 0.3%; '1' in 2.8%; '2a' in
204 10.0%; '2b' in 20.5%; '3' in 38.0%; '4' in 63.0%; '5' in 88.0%) (Tasser and Tappeiner
205 2005), we computed total percent cover of non-native species and of each phytosociological
206 group for each subplot.
207
208 All variables were log- (for continuous data) or arcsin- transformed (for percent data) to
209 ensure homoskedasticity and normality of error distributions, and assessed for differences as
210 a function of treatment by 2-way ANOVA with study area as a random factor and using
211 Tukey's Honest Significant Difference (HSD) post-hoc test. All analyses were carried out in
212 the SPSS 20.0 statistical environment (IBM Corporation 2011).
213
214 **4. Results**
215
216 All stands exhibited high tree densities (1000 to 2300 trees per hectare), with a large
217 variability within treatments but a thinning trend as time since coppicing increased (Figure 2).
218 Conversely, basal area, mean diameter, and volume increase with time since coppicing, up to
219 yields of 450 m^3 ha^{-1} in stands harvested >40 years before sampling. Seedling density was
220 highly variable, between 260 and 7000 per hectare, mostly originated from seed and

predominantly by ash (64%), but decreased with increasing stand age (Figure 3). Canopy cover declined with increasing stand age (83%, 78%, and 74% respectively in TR1, TR2 and TR0, $p < 0.01$ with mixed-model ANOVA), possibly due to a structural change from a dense coppice to a high-forest with larger but sparser trees.

In all treatments, alder occupied preferentially the dominant vegetation layer, while ash was found in all layers. However, frequent coppicing simplified vertical stand structure. Relative to TR1 and TR2, old coppices showed a differentiation in two distinct tree layers, the upper dominated by alder, and the lower by ash (Table 1). In both medium and recently coppiced stands, dominant trees were still competing with each other within the same vegetation layer (<15 m height), even after 20-30 years. Moreover, recently coppiced stands (TR1) had a lower herbaceous and higher bare soil cover, likely due to the use of machines to transport harvested woods in the stand (as tracks observed in the field clearly showed).

Soils were rich in organic carbon and showed presence of gley. TOC%, TN%, and C/N ratio varied in the range of 4.5 to 19.3, 0.36 to 1.59, and 11.4 to 14.7, respectively, but they did not differ significantly between treatments (Table 2). Therefore, we ruled out topographic or edaphic effects in determining understory species composition.

Recently coppiced stands (TR1) had a higher total species richness, and TR1 and TR2 a higher Shannon diversity, than undisturbed stands (Table 3). However, this did not result in an increased naturalness of plant species composition. In fact, the number of *Fraxinetalia* species (*Carex remota, Carex brizoides, Carex pendula, Impatiens noli-tangere, Ribes rubrum, Equisetum telmateja*) remained substantially low (3 species per plot) and unchanged

along the chronosequence, while their cover decreased significantly from 80% in TR0, to 65% in TR2, and 49% in TR1.

Conversely, the frequency and cover of non-native species were significantly higher in both medium and recently coppiced stands than in TR0 (Table 3). All 10 non-native species sampled in the study areas (*Acer negundo, Acer palmatum, Juglans regia, Ligustrum sinensis, Paulownia tomentosa, Robinia pseudoacacia, Lonicera japonica, Parthenocissus quinquefolia, Duchesnea indica, Oxalis fontana*) were found only in the two latter stages of the chronosequence (Appendix 1). These areas also showed a significantly higher number or ruderal nitrophilous species (*Urtica dioica, Alliaria petiolata, Geum urbanum, Geranium robertianum, Oplismenus undulatifolius, Galeopsis pubescens, Polygonum hydropiper*), which lower the biodiversity value of alder stands and whose cover was relatively high (10%) even 20-30 years after treatment. Also the number and cover of species more typically occurring in mown and fertilized meadows or in hydrophylous herb communities (*Poa trivialis, Rubus caesius, Humulus lupulus, Filipendula ulmaria, Lythrum salicaria*), rather than in alder forests, were higher in recently and medium vs. old coppices (36% and 41% in TR1 and TR2, 5% in TR0). The same was true also for shrub species (*Viburnum opulus, Cornus sanguinea, Corylus avellana, Crataegus monogyna, Euonymus europaeus, Ligustrum vulgare, Frangula alnus, Salix cinerea*), as they can take advantage from higher light levels in recently opened gaps. Shrub cover was still high (41%) after 20-30 years from coppicing, showing the magnitude and long lasting legacy of silvicultural treatments on stand structure and light conditions.

5. Discussion

5.1 Forest structure and dynamics

The yield of black alder in Europe at age 80 is between 500 and 1000 $m^3 ha^{-1}$ (Lockow 1995; Sopp 1974). Old coppice stands included in this study approached the lower end of this range. The average yield of black alder in the forest district where the study was carried out was 148 $m^3 ha^{-1}$ (Bertani et al. 2003), which is indicative of the relative rarity of undisturbed mature alder forests in the area. Contrary to mountain areas, management of coppices in lowland sites of the Po plain is still quite active, due to their high accessibility and the predominantly private ownership.

In our study, seedling density decreased with increasing stand age. This confirms that alder regeneration by seed is difficult in closed stands, as seedlings are very intolerant of shading and rank herbaceous vegetation (Latham and Blackstock 1998). Herbaceous competitors (e.g. *Carex* spp.) were more abundant soon after coppicing (see below) and may have prevented alder seedling development (McVean 1956). Should alder seedlings take advantage of the higher light levels from canopy opening, establishment would decline soon after coppicing (Ash and Barkham 1976). At the same time, intense coppicing with few or no standards favored ash regeneration, both from seed and from faster sprout growth. Ash seedlings and sprouts may grow very slowly if in shade for several years, but are able to respond with rapid growth when gaps appear in the canopy (Merton 1970; Tapper 1992, 1993; Gatsuk et al. 1980). Once in the canopy, ash is able to overgrow and suppress alder through competition for light (Tapper 1996).

Finally, biomass, mean tree size, and complexity of vertical structure (i.e., number of tree layers) declined with increasing coppicing frequency. These effects persisted up to 30 years

294 after treatment, and may result in a loss of habitats and lower spatial heterogeneity of
295 resources (Motta et al. 2015) that can hamper diversity and ecosystem stability. A more
296 diverse array of tree sizes and the existence of multiple vertical layers instead may provide a
297 greater number of potential ecological niches for a wide array of relevant forest biota, such as
298 invertabrates, birds, lichens (e.g. Müller et al. 2005; Czeszczewik and Walankiewicz 2006;
299 Nascimbene et al. 2013; Negro et al. 2015).

300

301 *5.2 Effects of coppice management on habitat conservation*

302 Many herb species typically occurring in woodland communities are perennials and can
303 persist throughout the coppice cycle. Under a coppice regime, the relatively constant and
304 saturated set of niches that is found in high forests is replaced by a variety of others, which
305 are filled by species capable of survival in a relatively wider range of ecological
306 circumstances (van der Werf 1991). Sometimes, the greatest threat to plant diversity is the
307 abandonment of coppicing, whereby many open-habitat species are slowly outshadowed
308 under the ever denser growing crowns (Baeten et al. 2009; Negro et al. 2015).

309

310 However, when the conservation target is the forest habitat, as in the case of black alder
311 remnants, too frequent coppicing can be detrimental. The first agent of damage is harvesting
312 machinery, which may induce compaction, rutting and stripping, irreversibly reduce soil
313 fertility and drainage, and alter species composition, especially on loamy soils (Decocq et al.
314 2005). Altered drainage may then produce habitat deterioration for other species of
315 conservation interest such as aquatic invertebrates, insects and amphibians (e.g., *Pelobates*
316 *fuscus insubricus*: Andreone et al. 2004). Second, frequent coppicing alters light and moisture
317 regimes, which facilitates ruderal, nitrophilous, tall herbs and non-native species (Funk et al.
318 2008) that can outcompete woodland species of interest. The post-coppicing communities

319 analyzed by this study contained species adapted to many types of canopy and soil
320 disturbance and relatively high light levels such as ruderal species, nitrophilous species (e.g.,
321 *Poa trivialis*, *Urtica dioica*, *Rubus caesius*) (Honnay et al. 1999; De Keersmaeker et al.
322 2004), tall herbs, and shrubs. In undisturbed stands, germination of these species is prevented
323 by low light levels and the abundant leaf litter (Sydes and Grime 1981). But if this vegetation
324 component increases due to the high coppicing frequency, it may progressively hamper
325 specialist woodland herbs by competitive exclusion (Hipps et al. 2005). Canopy removal
326 during coppicing may increase the amount of solar radiation reaching the soil, the quantity
327 and composition of light, the temperature, humidity, evaporation and mineralization rates.
328 Seeds of many open-habitat species require a high temperature regime for germination and it
329 is apparent that this occurs only when there is no vegetation present to cast shade, such as in
330 the first two years after coppicing. Moreover, the decrease of C and N in recently coppiced
331 stands might indicate an effect of coppicing frequency on organic matter recycling and
332 mineralization. Soils of undisturbed forests have been previously found to be richer in C and
333 N than in intensely harvested ones (Johnson and Curtis 2001; Finér et al. 2003), including in
334 alder carr (Honnay et al. 1999; Verheyen et al. 1999; Dzwonko 2001; Falkengren-Grerup et
335 al. 2006; Orczewska 2009). This may result from a more prolonged accumulation of litter
336 that, in the case of alder, has a high N content and rate of decomposition (Karkanis 1975;
337 Pereira et al. 1998). However, also the opposite was found, i.e., a higher N availability in
338 regularly harvested stands, due to a higher organic matter turnover rate (Covington 1981;
339 Kimmins 1987; Koerner et al. 1997; Keersmaeker et al. 2004). Canopy removal increases soil
340 temperature (Carlson and Groot 1997; Pennock and Kessel 1997; Hashimoto and Suzuki
341 2004), decreases soil water content (Ma et al. 2013) and shortens saturation periods, therefore
342 promoting a faster mineralization in harvested stands. Such different findings may depend on

343 regional climate (Yin et al. 1989), soil fertility, former agricultural use (Compton and Boone
344 2000), its duration, and time since abandonment.
345
346 The picture is further complicated by invasion of non-native species. Changes in the
347 hydrologic cycle of humid forests, induced e.g. by upstream water use, canopy cover
348 changes, or climate change, can further facilitate invasion by species adapted to drier
349 conditions (Huston 2004). Moreover, floodplain forests remnants can be more vulnerable to
350 plant invasions due to their small size and high perimeter-to-area ratio, which facilitates
351 species colonization from the margins and makes any canopy disturbance a potential threat to
352 native species. Among the non-native species found by this study in alder stands, some have
353 a well-documented high degree of invasiveness, e.g., *Acer negundo* L., *Robinia pseudoacacia*
354 L., *Lonicera japonica* Thunb. (Regione Piemonte 2015a), while for some others this is less
355 certain. However, also the latter ones have been previously reported in other areas of
356 Piedmont region, indicating that their naturalization potential and invasiveness are likely
357 higher than expected, e.g., *Ligustrum sinense* (Lonati et al. 2014; Soldano et al. 2015) or
358 *Paulownia tomentosa* (Selvaggi 2014). Once established, non-native species can persist due
359 to their high regeneration potential (also by vegetative reproduction), high degree of
360 adaptation to disturbances, allelopathy, and influx of non-native seeds in the soil seed bank
361 (Lorenzo et al. 2013; Gioria and Pyšek 2015).
362
363 For all these reasons, we recommend amending the current legislation and introducing
364 mandatory Implications Assessment procedures everywhere alder forests are susceptible to be
365 impacted by coppicing in the negative ways suggested by our analyses. Should coppice need
366 to be maintained, best silvicultural practices should include higher retention of living and
367 dead biomass, longer rotations (Della Rocca et al. 2014), and cessation of all drainage

activities, as a high water level inhibits the vigorous growth of expansive, nutrient-demanding species and reduces the competitive exclusion of woodland flora by such herbs. Promoting shadier conditions in the forest floor may also limit the expansion of ruderal and non-native plant species, and facilitate the immigration and establishment of typical woodland herbs (Orczewska 2009).

6. Conclusion

Many species and habitats associated with natural forested floodplains have disappeared from most of Europe. Current silvicultural practices, especially in small private woodlots, may increase pressures on biodiversity and ecosystem conservation. Our study showed that frequent coppicing has negative consequences for the conservation of black alder forests in the western part of the Po Plain (northern Italy), inducing a simplification of stand structure, a deterioration of species composition, and the spread of non-native plant species. Such negative effects persisted even 20-30 years after cutting, suggesting that the cumulative effects of repeated frequent coppicing would be characterized by even harsher impacts.

Acknowledgments

This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors. The authors would like to acknowledge Giulia Piancino and Emanuele Sibona for image analysis, and Emanuele Pintaldi for field sampling and soil analysis. The authors support the campaign #ricercaprecaria for the full implementation of the European Charter for Researchers and the acknowledgement of all researchers as workers, and the Change.org petition “Salviamo la ricerca italiana”

(https://www.change.org/p/salviamo-la-ricerca-italiana) for the increase of research funding in Italy to the levels requested by the EU Lisbon strategy.

Conflict of Interest

The authors declare that they have no conflict of interest.

Appendices

Additional supporting information in the online version of this article (see "Supplementary Material") contains the following: ESM_1 - List of native and non-native species, phytosociological optimum (according to Aeschimann et al. 2004), species frequency (%), and minimum and maximum cover (cover-abundance scores according to Braun-Blanquet 1932)

References

Aeschimann D, Lauber K, Moser MD, Theurillat JD (2004) Flora alpina. Zanichelli, Bologna

Andreone F, Eusebio Bergò P, Bovero S, Gazzaniga E (2004) On the edge of extinction? The spadefoot *Pelobates fuscus insubricus* in the Po Plain, and a glimpse at its conservation biology. Bollettino di Zoologia 71(S2):61-72

Andreone F, Eusebio Bergò P, Cerfolli F (2001). Studio di base per il Sito di Importanza Comunitaria (SIC) Laghi d'Ivrea. WWF Italia and Ministero dell'Ambiente, Servizio Conservazione Natura. http://master.wwf.it/UserFiles/File/AltriSitiWWF/Piemonte/documenti/piani_gestione/ivrea/2000_Studio%20di%20base_pagg_01_69.pdf. Accessed 22 September 2015

Ash JE, Barkham JP (1976) Changes and variability in the field layer of a coppiced woodland in Norfolk, England. J Ecol 64:697-712

Baeten L, Bauwens B, De Schrijver A, De Keersmaeker L, Van Calster H, Vandekerkhove K, Roelandt B, Beeckman H, Verheyen K (2009) Herb layer changes (1954-2000) related to the conversion of coppice-with-standards forest and soil acidification. Appl Veg Sci 12(2):187-197

Bertani R, Quaglia M, Ghepardi ME, Giandrini R, Chiariglione D, Biorcio C, Galante S, Santoro A, Innocenti M, Doglione A, Oradini A, Bazzano V, Longo F, Albery R, Bonaria M, Balangione GL (2003) Area forestale: Canavese – Eporediese. Piano Forestale Territoriale. Regione Piemonte, Torino

Bond G, Fletcher W, Ferguson T (1954) The development and function of the root nodules of *Alnus*, *Myrica* and *Hippophae*. Plant Soil 5:309-323

Braun-Blanquet J (1932) Plant sociology. McGraw-Hill, New York

Brown AG, Harper D, Peterken GF (1997) European floodplain forests: structure, functioning and management. Global Ecol Biogeogr Lett 1:169-178

Carlson DW, Groot A (1997) Microclimate of clear-cut, forest interior, and small openings in trembling aspen forest. Agr For Meteorol 87(4):313-329

Carraro F, Medioli F, Petrucci F (1975) Geomorphological study of the morainic amphitheatre of Ivrea, Northwest Italy. Bull R Soc New Zealand 13:89-93

Cech TL (1998) *Phytophthora* decline of alder (*Alnus* spp.) in Europe. J Arboriculture (USA) 24:339-343

Celesti-Grapow L, Alessandrini A, Arrigoni PV, Banfi E, Bernardo L, Bovio M, Brundu G, Cagiotti MR, Camarda I, Carli E, Conti F, Fascetti S, Galasso G, Gubellini L, La Valva V,

Lucchese F, Marchiori S, Mazzola P, Peccenini S, Poldini L, Pretto F, Prosser F, Siniscalco C, Villani MC, Viegi L, Wilhalm T, Blasi C (2009) The inventory of the non-native flora of Italy. Plant Biosystems 143:386-430

Claessens H (2003) The alder populations of Europe. For Comm Bull 126:5-14

Claessens H, Oosterbaan A, Savill P, Rondeux J (2010) A review of the characteristics of black alder (*Alnus glutinosa* (L.) Gaertn.) and their implications for silvicultural practices. Forestry 83:163-175

Compton JE, Boone RD (2000) Long-term impacts of agriculture on soil carbon and nitrogen in New England forests. Ecology 81(8):2314-2330

Cools N, De Vos B (2010) Sampling and Analysis of Soil. Manual Part X. In: Manual on methods and criteria for harmonized sampling, assessment, monitoring and analysis of the effects of air pollution on forests. UNECE, ICP Forests, Hamburg

Covington WW (1981) Changes in forest floor organic matter and nutrient content following clear cutting in northern hardwoods. Ecology 62(1):41-48

Czeszczewik D, Walankiewicz W (2006) Logging affects the white-backed woodpecker *Dendrocopos leucotos* distribution in the Białowieza Forest. Ann Zool Fenn 43:221–227

De Keersmaeker L, Martens L, Verheyen K, Hermy M, De Schrijver A, Lust N (2004) Impact of soil fertility and insolation on diversity of herbaceous woodland species colonizing afforestations in Muizen forest (Belgium). For Ecol Manage 188(1):291-304

Decocq G, Aubert M, Dupont F, Bardat J, Wattez-Franger A, Saguez R, De Foucault B, Alard D, Delelis-Dusollier A (2005) Silviculture-driven vegetation change in a European temperate deciduous forest. Ann For Sci 62(4):313-323

461 Deiller AF, Walter JM, Trémolières M (2003) Regeneration strategies in a temperate
462 hardwood floodplain forest of the Upper Rhine: sexual versus vegetative reproduction of
463 woody species. For Ecol Manage 180(1):215-225

464 Della Rocca F, Stefanelli S, Pasquaretta C, Campanaro A, Bogliani G (2014) Effect of
465 deadwood management on saproxylic beetle richness in the floodplain forests of northern
466 Italy: some measures for deadwood sustainable use. J Insect Conserv 18(1):121-136

467 Dethioux M (1974) Quelques éléments de l'écologie du semis de l'aulne glutineux. Ardenne
468 et Gaume 24(3):118-129

469 Douda J, Čejková A, Douda K, Kochánková J (2009) Development of alder carr after the
470 abandonment of wet grasslands during the last 70 years. Ann For Sci 66(7):1-13

471 Dzwonko Z (2001) Assessment of light and soil conditions in ancient and recent woodlands
472 by Ellenberg indicator values. J Appl Ecol 38(5):942-951

473 EEA (2012) 91E0 Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (*Alno-Padion*,
474 *Alnion incanae, Salicion albae*). Report under the Article 17 of the Habitats Directive.
475 European Environment Agency.
476 http://bd.eionet.europa.eu/article17/reports2012/static/factsheets/forests/91e0-alluvial-forests-
477 with-alnus-glutinosa-and-fraxinus-excelsior-.pdf. Accessed 11 April 2016

478 Ellenberg H (1996) Vegetation Mitteleuropas mit den Alpen in ökologischer, dynamischer
479 und historischer Sicht. Struttgart, Ulmer.

480 Falkengren-Grerup U, ten Brink DJ, Brunet J (2006) Land use effects on soil N, P, C and pH
481 persist over 40–80 years of forest growth on agricultural soils. For Ecol Manage 225(1):74-81

482 Finér L, Mannerkoski H, Piirainen S, Starr M (2003) Carbon and nitrogen pools in an old-
483 growth, Norway spruce mixed forest in eastern Finland and changes associated with clear-
484 cutting. For Ecol Manage 174(1):51-63

485 Funk DT (1990) *Alnus glutinosa* (L.) Gaertn. European alder. In: Burns RM, Honkala BH
486 (eds) Silvics of North America, Hardwoods. USDA Forest Service, Washington DC, pp.105-
487 115.

488 Funk JL, Cleland EE, Suding KN, Zavaleta ES (2008) Restoration through reassembly: plant
489 traits and invasion resistance. Trends Ecol Evol 23(12):695-703

490 Gatsuk LE, Smirnova OV, Vorontzova LI, Zaugolnova LB, Zhukova LA (1980) Age states
491 of plants of various growth forms: a review. J Ecol 68:675-696

492 Giardina C, Huffmans S, Binkley D, Caldwell B (1995) Alders increase soil phosphorus
493 availability in a Douglas-fir plantation. Can J For Res 25:1652-1657

494 Gioria M, Pyšek P (2015) The legacy of plant invasions: changes in the soil seed bank of
495 invaded plant communities. BioScience 66(1): 40-53

496 Hashimoto S, Suzuki M (2004) The impact of forest clear-cutting on soil temperature: a
497 comparison between before and after cutting, and between clear-cut and control sites. J For
498 Res 9(2):125-132

499 Hipps NA, Davies MJ, Dodds P, Buckley GP (2005) The effects of phosphorus nutrition and
500 soil pH on the growth of some ancient woodland indicator plants and their interaction with
501 competitor species. Plant Soil 271(1-2):131-141

502 Honnay O, Hermy M, Coppin P (1999) Effects of area, age and diversity of forest patches in
503 Belgium on plant species richness, and implications for conservation and reforestation. Biol
504 Cons 87(1):73-84

505 Huang LK, Wang, MJJ (1995) Image thresholding by minimizing the measure of fuzziness.
506 Pattern Recognition 28(1):41-51

507 Huntley B, Birks HJB (1983) An atlas of past and present pollen maps for europe: 0–13 000
508 years ago. Cambridge University Press, Cambridge

509 Huston MA (2004) Management strategies for plant invasions: manipulating productivity,
510 disturbance, and competition. Divers Distrib 10(3):167-178

511 IBM Corporation (2011) IBM SPSS Statistics for Windows, Version 20.0. IBM Corp,
512 Armonk NY

513 Johnson DW, Curtis PS (2001) Effects of forest management on soil C and N storage: meta
514 analysis. For Ecol Manage 140(2):227-238

515 Johnson MRW (1973) Displacement on the Insubric Line. Nature 241(110):116-117

516 Kajba D, Gracan J (2003) EUFORGEN technical guidelines for genetic conservation and use
517 for Black Alder (*Alnus glutinosa*). International Plant Genetic Resources Institute, Rome

518 Kapustinskaite T (1960) Natural regeneration in *Alnus glutinosa* stands in Lithuania, and
519 ways of improving it. Lietuvos Misku Ukio Mokslu Tyrimo Instituto Darbai 5:89-152

520 Karkanis M (1975) Decomposition of litter of various species of deciduous trees and its effect
521 on soil environment. Fragmenta Floristica et Geobotanica 21:71-97

522 Kimmins JP (1987) Forest ecology. Macmillan, New York

523 Koerner W, Dupouey JL, Dambrine E, Benoit M (1997) Influence of past land use on the
524 vegetation and soils of present day forest in the Vosges mountains, France. J Ecol 85:351-358

525 Koop H (1987) Vegetative reproduction of trees in some European natural forests. Vegetatio
526 72(2):103-110

527 Kremer F, van der Stegen J, Gafo Gomez-Zamalloa M, Szedlak T (2015) Natura 2000 and
528 forests. Office for Official Publication of European Communities, Luxembourg

Latham J, Blackstock TH (1998) Effects of livestock exclusion on the ground flora and regeneration of an upland *Alnus glutinosa* woodland. Forestry 71:191–197

Li CH, Tam PKS (1998) An iterative algorithm for minimum cross entropy thresholding. Pattern Recognition Lett 18(8):771-776

Lockow KW (1995) Die neue Ertragtafel für Roterle – Modellstruktur und Anwendung in der Forstpraxis. Beiträge fûr Forstwirtschaft und Landschaftsökologie 29(2):49-55

Lonati M, Meloni F, Vacchiano G, Ferrarato M (2014) *Ligustrum sinense* Lour. (*Oleaceae*). In: Selvaggi A, Soldano A, Pascale M, Dellavedova R (eds) Note Floristiche Piemontesi 545-604. Riv Piem St Nat 35:401

Lonati M, Vacchiano G, Berretti R, Motta R (2013) Effect of stand-replacing fires on Mediterranean plant species in their marginal alpine range. Alp Bot 123:123-133

Lorenzo P, Hussain MI, González L (2013) Role of allelopathy during invasion process by alien invasive plants in terrestrial ecosystems. In: Cheema ZA, Farooq M, Wahid A (eds) Alleopathy. Springer, New York, pp. 3-21

Ma JZ, He JH, Qi S, Zhu G, Zhao W, Edmunds WM, Zhao Y (2013) Groundwater recharge and evolution in the Dunhuang Basin, northwestern China. Appl Geochem 28:19–31

McVean DN (1953) Biological flora of the British Isles: *Alnus glutinosa* (L.) Gaertn. J Ecol 41: 447–466

McVean DN (1955) Ecology of *Alnus glutinosa* (L.) Gaertn. II Seed distribution and germination. J Ecol 43:61–71

McVean DN (1956) Ecology of *Alnus glutinosa* (L.) Gaertn. III Seedling establishment. J Ecol 44:195–218

551 Merton LFH (1970) The history and status of the woodlands of the Derbyshire limestone. J
552 Ecol 58:723-744

553 Minuzzo C, Tisi A, Caramiello R, Siniscalco C (2005) Flora acquatica e palustre della zona
554 dei “Cinque Laghi” di Ivrea. Riv Piem St Nat 26:41-71

555 Moiroud A (1991) La symbiose fixatrice d’azote. Forêt Entreprise 75:18-26

556 Motta R, Garbarino M, Berretti R, Meloni F, Nosenzo A, Vacchiano G (2015) Development
557 of old-growth characteristics in uneven-aged forests of the Italian Alps. Eur J For Res
558 134:19-31

559 Müller J, Strätz C, Hothorn T (2005) Habitat factors for land snails in European beech forests
560 with a special focus on coarse woody debris. Eur J For Res 124(3):233-242.

561 Nascimbene J, Thor G, Nimis PL (2013) Effects of forest management on epiphytic lichens
562 in temperate deciduous forests of Europe – A review. For Ecol Manage 298:27–38.

563 Natura 2000 Network Wiewer (2016) NATURA 2000 – Standard data form. IT1110021 -
564 Laghi di Ivrea. http://natura2000.eea.europa.eu/Natura2000/SDF.aspx?site=IT1110021.
565 Accessed 29 March 2016.

566 Negro M, Vacchiano G, Berretti R, Chamberlain DE, Palestrini C, Motta R, Rolando A
567 (2015) Effects of forest management on ground beetle diversity in alpine beech (*Fagus*
568 *sylvatica* L.) stands. For Ecol Manage 328:300-309

569 Orczewska A (2009) The impact of former agriculture on habitat conditions and distribution
570 patterns of ancient woodland plant species in recent black alder (*Alnus glutinosa* (L.) Gaertn.)
571 woods in south-western Poland. For Ecol Manage 258(5):794-803

572 Orlandi S, Probo M, Sitzia T, Trentanovi G, Garbarino M, Lombardi G, Lonati M (2016)
573 Environmental and land use determinants of grassland patch diversity in the western and
574 eastern Alps. Biodiv Cons 25:275–293

575 Otsu N (1979) A threshold selection method from gray-level histograms. IEEE Trans Sys
576 Man Cyber 9:62-66

577 Pennock DJ, Kessel CV (1997) Clear-cut forest harvest impacts on soil quality indicators in
578 the mixedwood forest of Saskatchewan, Canada. Geoderma 75:13–32

579 Pereira AP, Graça MAS, Molles M (1998) Leaf litter decomposition in relation to litter
580 physico-chemical properties, fungal biomass, arthropod colonization, and geographical origin
581 of plant species. Pedobiologia 42:316-327.

582 Peterjohn W, Correll D (1984) Nutrient dynamics in an agricultural watershed: observations
583 on the role of a riparian forest. Ecology 65:1466-1475

584 Piazzi M, Boni I, Petrella F, Martalò PF (2007) La carta dei suoli del Piemonte a scala
585 1:250.000 con note illustrative e CD. Selca, Firenze

586 Piégay H, Pautou G, Ruffioni C (2003) Les forêts riveraines des cours d'eau. Ecologie,
587 fonctions et gestion. Institut pour le Développement Forestier, Paris

588 Pittarello M, Probo M, Lonati M, Lombardi G (2016) Restoration of sub-alpine shrub-
589 encroached grasslands through pastoral practices: Effects on vegetation structure and
590 botanical composition. Appl Veg Sci, in press. doi: 10.1111/avsc.12222

591 Pokorný P, Klimešová J, Klimeš L (2000) Late Holocene history and vegetation dynamics of
592 a floodplain alder carr: a case study from eastern Bohemia, the Czech Republic. Folia Geobot
593 Phytotax 35:43–58

Regione Piemonte (2015a) Le specie esotiche invasive. http://www.regione.piemonte.it/ambiente/tutela_amb/esoticheInvasive.htm. Accessed 1 September 2015

Regione Piemonte (2015b) Regolamento Regionale recante: "Regolamento forestale di attuazione dell'articolo 13 della Legge Regionale 10 febbraio 2009, n. 4 (Gestione e promozione economica delle foreste). Testo integrato. http://www.regione.piemonte.it/foreste/images/files/pian_gest/dwd/nuova_legge/testointegrato2015.pdf. Accessed 11 April 2016

Regione Piemonte (2016a) Autorizzazioni e comunicazioni forestali art.14 L.R. 4/2009. http://www.sistemapiemonte.it/ambiente/sipap/accesso_aut_forestali.shtml. Accessed 10 April 2016

Regione Piemonte (2016b) Misure di conservazione per la tutela della Rete Natura 2000 del Piemonte. Testo coordinato. http://www.regione.piemonte.it/parchi/cms/dwd/MdC_testo_coord_2016.pdf. Accessed 11 April 2016

Richardson DM, Holmes PM, Esler KJ, Galatowitsch SM, Stromberg JC, Kirkman SP, Pysek P, Hobbs RJ (2007) Riparian vegetation: degradation, alien plant invasions, and restoration prospects. Divers Distrib 13:126–139

Schindelin J, Arganda-Carreras I, Frise E, Kaynig V, Longair M, Pietzsch T, Preibisch S, Rueden C, Saalfeld S, Schmid B, Tinevez JY, White DJ, Hartenstein V, Eliceiri K, Tomancak P, Cardona A (2012), Fiji: an open-source platform for biological-image analysis. Nature Methods 9(7):676-682

Schnitzler A (1994) European alluvial hardwood forests of large floodplains. J Biogeogr 21:604–623

618 Selvaggi A (2014) *Paulownia tomentosa* (Sprengel) Steudel (*Paulowniaceae*). In: Selvaggi
619 A, Soldano A, Pascale M, Dellavedova R (eds) Note Floristiche Piemontesi 604-705. Riv
620 Piem St Nat 36:404-405

621 Shaw K, Roy S, Wilson B (2014) *Alnus glutinosa*. The IUCN Red List of Threatened Species
622 2014: e.T63517A3125479. http://dx.doi.org/10.2305/IUCN.UK.2014-
623 3.RLTS.T63517A3125479.en. Accessed 11 April 2016

624 Soldano A, Bouvet D, Viñals N (2015) *Ligustrum sinense* Lour. (*Oleaceae*). In: Selvaggi A,
625 Soldano A, Pascale M, Dellavedova R (eds) Note Floristiche Piemontesi 604-705. Riv Piem
626 St Nat 36:328

627 Sopp L (1974) Fatömeg – szamitazi tablazatok. Mezögazdasagi Kiado, Budapest

628 Sydes C, Grime JP (1981) Effects of tree leaf litter on herbaceous vegetation in deciduous
629 woodland. II. Experimental investigation. J Ecol 69:249-262

630 Tapper PG (1992) Demography of persistent juveniles in Fraxinus excelsior. Ecography
631 15:385-392

632 Tapper PG (1993) The replacement of *Alnus glutinosa* by *Fraxinus excelsior* during
633 succession related to regenerative differences. Ecography 16:212-218

634 Tapper PG (1996) Tree dynamics in a successional *Alnus—Fraxinus* woodland. Ecography
635 19:237-244

636 Tasser E, Tappeiner U (2005) New model to predict rooting in diverse plant community
637 compositions. Ecol Model 185:195-211

638 Van der Werf S (1991) Bosgemeenschappen. Natuurbeheer in Nederland deel 5. Pudoc,
639 Wageningen

640 Verheyen K, Bossuyt B, Hermy M, Tack G (1999) The land use history (1278–1990) of a
641 mixed hardwood forest in central Belgium and its relationship with chemical soil
642 characteristics. J Biogeogr 26:1115–1128

643 Wolf A, Møller PF, Bradshaw RHW, Bigler J (2004) Storm damage and long-term mortality
644 in a semi-natural, temperate deciduous forest. For Ecol Manage 188:197–210

645 Yen JC, Chang FJ, Chang S (1995) A new criterion for automatic multilevel thresholding.
646 IEEE Trans Image Processing 4(3):370-378

647 Yin X, Perry JA, Dixon RK (1989) Influence of canopy removal on oak forest floor
648 decomposition. Can J For Res 19:204–214

649

Tables

Table 1. Percent cover (mean ± standard error) of different vegetation layers (all species, black alder and ash) by treatment. Different letters indicate significant differences between treatments at $p < 0.10$ (ANOVA with Tukey's HSD test).

% cover of vegetation layer	TR1 (10-15 years)		TR2 (20-30 years)		TR0 (>40 years)	
Upper tree	0 ± 0.0	a	0 ± 0.0	a	82 ± 2.3	b
Lower tree	83 ± 1.8	b	87 ± 1.2	b	19 ± 1.7	a
Upper shrub	21 ± 4.2	b	16 ± 2.1	ab	10 ± 1.4	a
Lower shrub	41 ± 3.1	c	26 ± 2.4	b	14 ± 1.4	a
Herbaceous	78 ± 3.4	a	93 ± 0.9	b	91 ± 1.1	b
Bare soil	22 ± 3.4	b	7 ± 0.9	a	9 ± 1.1	a
Alnus glutinosa						
Total	45 ± 3.0	a	61 ± 4.5	b	78 ± 3.3	c
Upper tree	0 ± 0.0	a	0 ± 0.0	a	78 ± 3.3	b
Lower tree	45 ± 3.0	b	61 ± 4.5	c	0 ± 0.0	a
Upper shrub	0 ± 0.3	a	0 ± 0.0	a	0 ± 0.0	a
Lower shrub	-		-		-	
Fraxinus excelsior						
Total	66 ± 5.5	b	55 ± 6.7	ab	48 ± 3.7	a
Upper tree	0 ± 0.0	a	0 ± 0.0	a	3 ± 1.6	b
Lower tree	44 ± 4.7	b	30 ± 5.0	a	21 ± 2.2	a
Upper shrub	18 ± 3.5	a	16 ± 2.2	a	11 ± 1.4	a
Lower shrub	4 ± 1.2	a	9 ± 1.6	b	12 ± 1.3	b

Table 2. Soil data by treatment in the study area (means ± standard error). Different letters indicate significant differences between treatments at $p < 0.10$ (ANOVA with Tukey's HSD test).

Variable	TR1 (8-10 years)	TR2 (20-30 years)	TR0 (>40 years)
TN%	0.8 ± 0.08 a	0.9 ± 0.07 a	1.0 ± 0.07 a
TOC%	10.5 ± 1.05 a	11.8 ± 0.84 a	13.0 ± 0.81 a
C/N	12.6 ± 0.24 a	12.8 ± 0.17 a	12.9 ± 0.2 a

673
674 **Table 3**. Diversity, richness, and cover of functional groups (mean ± standard error) by
675 treatment. Different letters indicate significant differences between treatments at $p < 0.05$
676 (ANOVA with Tukey's HSD test).
677

% cover of vegetation layer	TR1 (10-15 years)		TR2 (20-30 years)		TR0 (>40 years)	
Total richness	15.4 ± 0.50	b	12.7 ± 0.67	a	11.3 ± 0.69	a
Shannon index (H)	3.0 ± 0.06	c	2.7 ± 0.09	b	2.1 ± 0.05	a
Species number						
Non-native species	1.7 ± 0.27	c	0.9 ± 0.19	b	0.0 ± 0.00	a
Ruderal species	1.3 ± 0.12	b	1.1 ± 0.19	b	0.5 ± 0.13	a
Hydrophilous tall herb species	2.3 ± 0.21	b	3.1 ± 0.22	b	2.1 ± 0.09	a
Shrub species	2.8 ± 0.31	b	1.4 ± 0.21	a	1.9 ± 0.36	a
Fraxinetalia species	3.0 ± 0.22	a	2.9 ± 0.19	a	3.0 ± 0.24	a
Percent cover						
Non-native species	15 ± 2.0	c	4 ± 1.2	b	0 ± 0.0	a
Ruderal species	15 ± 2.4	c	10 ± 2.4	b	0 ± 0.2	a
Tall herb species	36 ± 4.6	b	41 ± 4.6	b	5 ± 1.0	a
Shrub species	16 ± 2.4	b	6 ± 1.4	a	3 ± 0.9	a
Fraxinetalia species	49 ± 3.4	a	65 ± 3.3	b	80 ± 2.9	c

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682 **Figure captions**

683

684 **Fig. 1** Location of the study areas in Piedmont, Italy (left: blue – Special Protection Areas,
685 red – Sites of Community Importance, green – Ramsar sites) and within the Site of
686 Community importance “Laghi di Ivrea” (right).

687

688 **Fig. 2** Stand structural variables in the study areas by treatment. TR0: old (>40 years), TR1:
689 medium (10-20 years), TR2: recent coppice (20-30 years). Different letters indicate
690 significant differences between treatments at $p < 0.10$ (Mann-Whitney test).

691

692 **Fig. 3** Regeneration (individuals per hectare) in the study areas by treatment. TR0: old (>40
693 years), TR1: medium (10-20 years), TR2: recent coppice (20-30 years). Different letters
694 indicate significant differences between treatments at $p < 0.10$ (Mann-Whitney test).

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revised Figure 1

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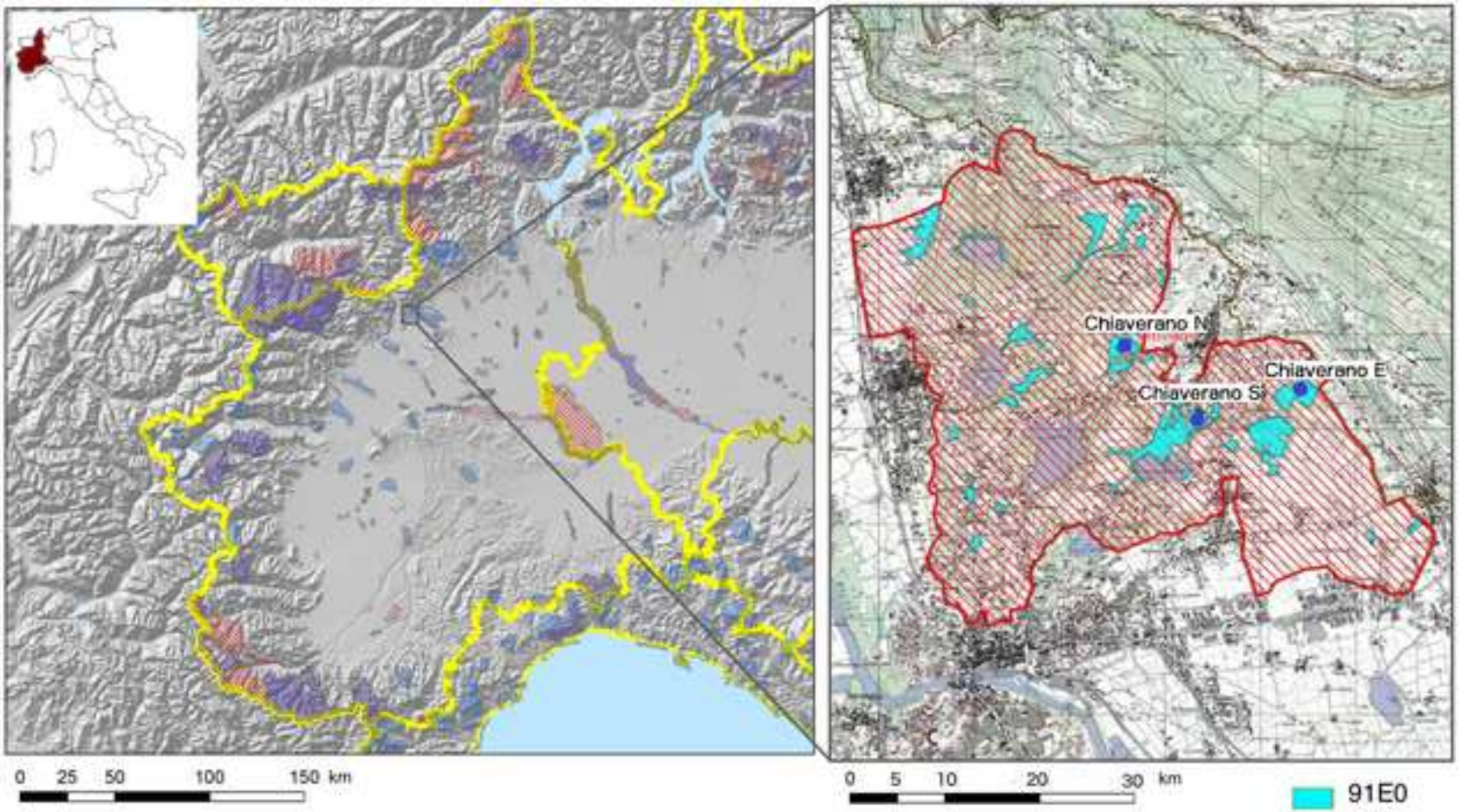

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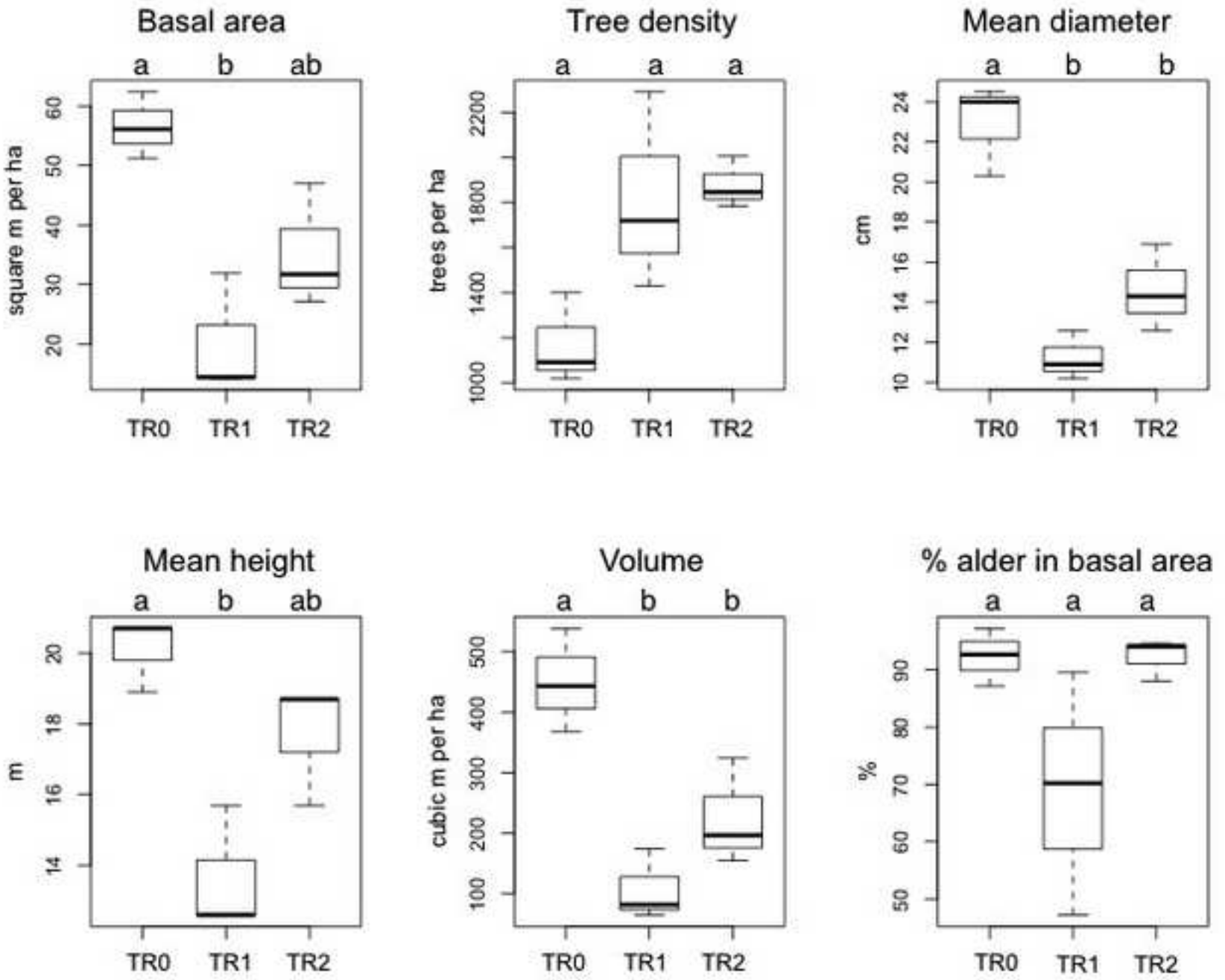

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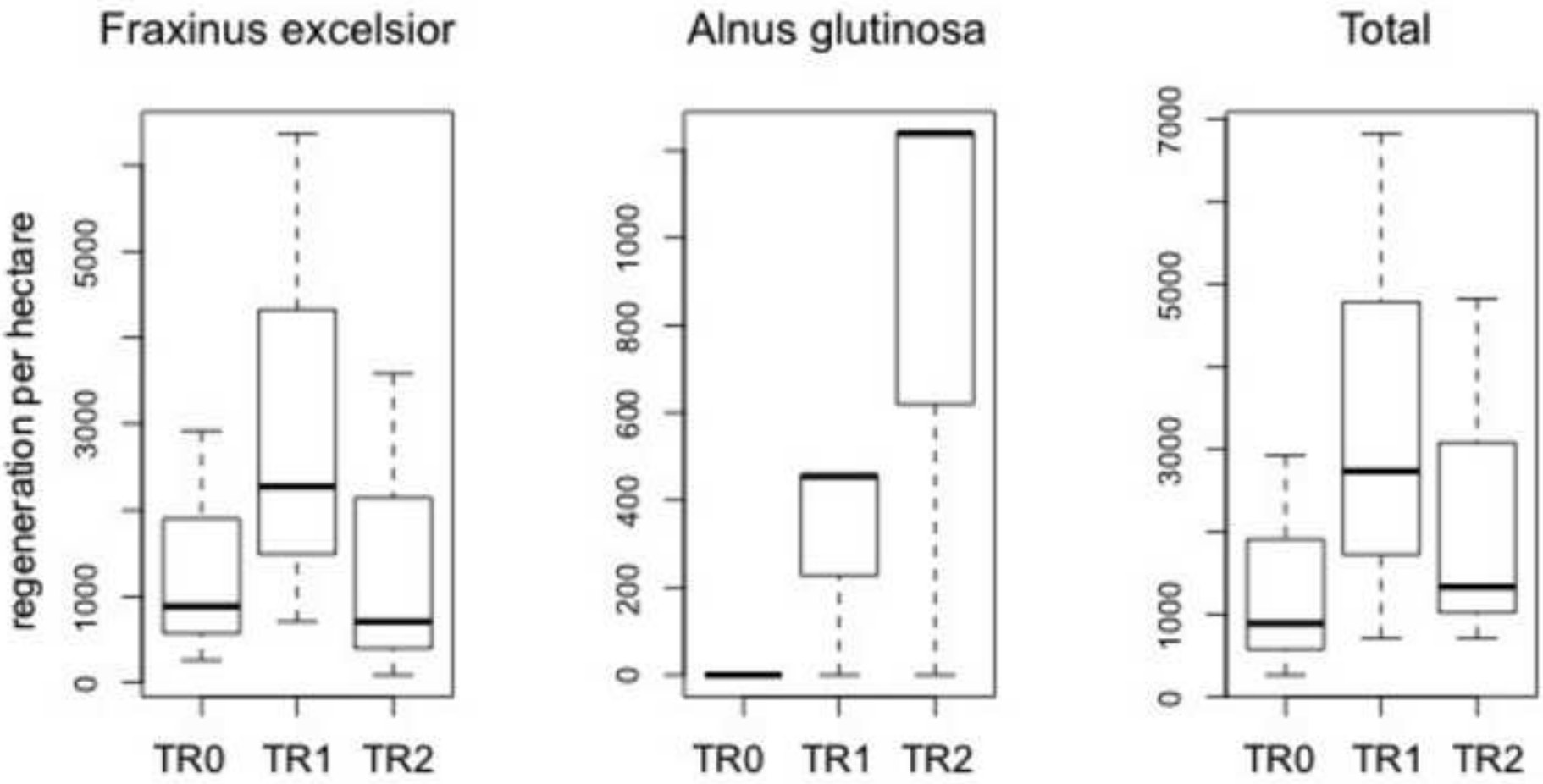

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